Response of vegetation to raising the water table as part of the Exmoor Mire Restoration Project

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Many factors influence vegetation community composition. Hydrological variables, particularly water table height, greatly affect mires, which are globally threatened habitats in need of conservation. On Exmoor, previous drainage caused the loss of wetland vegetation, but it is being restored by rewetting the bogs. Previous studies have shown such restoration to be successful in re-establishing healthy mire communities, especially *Sphagnum* mosses. Here, a survey was conducted to assess the current condition of the vegetation post-rewetting. Dipwells were installed to analyse the water table height and how this affects vegetation composition, particularly the mosses and the graminoids. It was found that while the community has remained dominated by *Molinia caerulea*, overall species richness and abundance have increased, and so have those of the mosses, whereas the graminoids have showed decline. Both groups were present at all water table levels but in areas which had the highest depths, the graminoids weren’t as abundant. This suggests that the water table is sufficiently high to support recolonisation of mosses but in order to strengthen this trend and discourage graminoids more pools would be beneficial. The project is only in its early stages and continued monitoring is needed to evaluate the eventual outcome of restoration.

**Keywords:** Exmoor; mire restoration; water table; mosses; graminoids; ecohydrology

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1. INTRODUCTION

Many biotic and abiotic factors determine the distribution and survival success of plants. The main biotic influences are predation from higher trophic organisms and competition from other plants. Of the environmental factors, climate and topography are important as well as the physical and chemical characteristics of the growing substrate. For a given species, the correct chemical balance of nutrients and pH must be present in order for survival. The key physical attribute of a growing substrate is moisture content and different species of plant are distinctively adapted to differential soil moisture conditions. This directly relates to the height of the water table on top of which the plant is growing. Thus, vegetation communities, and also whole ecosystems, can be studied in terms of hydrological regimes, which is essentially the principle behind the science of ecohydrology (Baird, 1999). Therefore, when the hydrology of a habitat is changed, for example from drainage or flooding, the vegetation responds and often a whole new community develops.

One of the many habitats that have been subject to hydrological changes is mire. These are wetland ecosystems characterised by stores of peat and a rich variety of specialised water-loving flora and fauna. Covering approximately 3% of the earth’s land surface (Carpenter et al., 2005), they provide humans with many important ecosystem services including carbon sequestration and hydrological regulation, as well as hosting a distinctive range of biodiversity. However, many of these mire habitats and their associated values are globally under threat from anthropogenic influences in the form of peat extraction for fuel and horticultural purposes, and drainage for agriculture (Tuittila et al., 2000). The major environmental consequences of these actions have been the conversion of what was once a sink of carbon into a source of carbon due to an increase in the aerobic decay of peat, and a local change in biodiversity as it has adapted to a lowered water table and the associated conditions of low nutrients and anaerobic soil/peat and acidic pH (Brooks & Stoneman, 1997). Natural recolonisation takes place by stress-tolerant vascular plants (Ferland & Rochefort, 1997). Fortunately, their value is now being recognised and conserved by mire restoration programmes around the world.

One type of mire is blanket bog. A priority habitat under the European Commission’s Habitats Directive, it has a limited global distribution requiring a cool wet climate so that peat develops extensively across the landscape. Fed solely by precipitation, this habitat is comprised mostly of Sphagnum mosses with other species, commonly Eriophorum species,
Trichophorum cespitosum, and Erica tetralix (Brooks & Stoneman, 1997). Blanket bog covers much of Exmoor and this moorland is no exception to the degradation described above. This has resulted in the once pristine mireland becoming dominated by grass and sedge species, predominantly Molinia caerulea, to the exclusion of other plants (Exmoor National Park Authority, 2007). The first important step in mire restoration is raising the water table (Malson et al., 2008) and this is followed by recolonisation of bog-associated plants, notably Sphagnum species (Vasander et al., 2003; Tuitiulla et al., 2000). The Exmoor Mire Restoration Project was established in 1998 and has to date rewetted, and hence restored, over 300 hectares of ‘Site of Special Scientific Interest’ (SSSI) moorland at 17 locations by blocking old drainage ditches with wood, peat and ditch-spoil bale dams. These form pools which store water, keeping it on the moor instead of flowing away. This raises the water table at the given location which then encourages blanket bog plant species to recolonise.

The type of vegetation in an area can be described and categorised using the National Vegetation Classification (NVC) system whereby the composition of vegetation is examined and fitted to a model. Using the classification for mires and heaths (Elkington et al., 2001), distinct types of mires can be seen. Previously, much of Exmoor, including the site studied here, had been classified as M25 Molinia caerulea-Potentilla erecta mire, which states that M. caerulea is very abundant and the rest of the vegetation is poor and made up mostly of rushes. The aim of the restoration project is to shift this vegetation type to a ‘healthier’ type of mire, the first successional stage being M2/M3 Sphagnum-dominated bog pools and eventually M17 Scirpus (Trichophorum) cespitosus – Eriophorum vaginatum blanket mire (David Smith, personal communication) which also has high abundances of Sphagnum moss.

The typically hostile conditions of mires prove fatal for many species of plants (Brix & Sorrell, 1996) and those that live there possess physiological, anatomical and morphological adaptations in order to survive. In particular, it is known that mire vegetation is strongly influenced by hydrological variables such as water table height, variability and surface water flow (Yabe & Onimaru, 1997). Here, the effects of a water table height gradient are studied. Different types of mire plant exploit different niches. Species that inhabit the ‘hummocks’ (drier areas) of a typical mire are differently adapted to species found in the ‘hollows’ (waterlogged areas) (Nordbakken, 1996). A high water table,
particularly above-ground poses the problem of waterlogging. The primary problem associated with waterlogging is a decrease in oxygen availability around the roots and this can disrupt many vital metabolic processes, for example ATP production (Losch & Busch, 2000), and it can also induce the production of phytotoxic compounds (Jones & Etherington, 1970). There are many strategies that plants employ in order to counteract this problem, for example they may ventilate underground waterlogged tissues by gas transport from within the shoots (Armstrong et al., 2001 in Brix & Sorrell, 1996), as well as accumulating carbohydrates to allow tissues to respire anaerobically (Brix & Sorrell, 1996).

The bryophytes (comprising of the liverworts, hornworts and mosses), are perhaps the best adapted to high moisture levels. Possessing no roots, water intake is conducted by capillary action in specialised dead cells, hydroids (Bell & Hemsley, 2000). These act as a water store and can in fact hold up to 20 times a moss’s dry weight (Richardson, 1981). Due to their reliance on continually wet substrate (Thompson & Waddington, 2008), mosses, especially *Sphagnum* species, can be used as key indicators of water table height.

On Exmoor, studies of the relationship between bryophytes and water table height and variability since rewetting have been conducted at three sites within the surrounding 275ha of the study site used here (Hand, 2009). It was found that overall species richness has increased while the richness of bryophytes increased between 2006 and 2008 but then decreased between 2008 and 2009. It was also found that as water table rose, there was an increase in the bryophytes that are highly associated with water, notably *Sphagnum*’s. There is a need to extend this research across other sites on the moor in order to see if the restoration project is being successful everywhere. It is also beneficial to study the relationship with water so that it can be seen what level of water, if any, has the greatest local biodiversity and therefore the level of water table to aim for in future rewetting. The previous study has only investigated bryophyte association with water, but this study focuses on the grasses, sedges and rushes (collectively the graminoids) as indicative species of water table change, as well as the bryophytes.

The questions that this project sets out to investigate are:

- How does the overall vegetation community respond to differing water table heights?
- How do target groups of species (the mosses and the graminoids) respond to differing water table heights?
Has restoration had the intended impact of increasing biodiversity, in particular increasing ‘healthy’ blanket bog species such as *Sphagnum* mosses whilst reducing the dominance of species associated with a degraded mire habitat?

2. MATERIAL AND METHODS

(a) Site description

The study was undertaken along the *Blackpitts 2* transect, (GB SS 762.6 422.0), part of the Unit 83 North Exmoor SSSI (275ha), of which Blackpitts is an area (see appendix 1i & ii for location maps). The area is 435m above sea level and has an average annual temperature of 7.5-11 °C and average annual rainfall of 1750mm. It is an area of upland blanket peat, comprising of a generally thin layer of peat (0.5-1m depth) with patches of thicker peat. The transect itself is on a plateau, part of a ‘col’ between 2 upland peaks and is technically a ‘saddleback mire’. Centuries of drainage caused the moorland vegetation here to become predominantly NVC type M25 *Molinia caerulea*- *Potentilla erecta* mire as well as degraded M17 *Scirpus cespitosus*- *Eriophorum vaginatum* blanket mire in some areas. Restoration works at Blackpitts, which has included ditch blocking using wooden, peat and ditch-spoil dams to retain water, were completed in Spring 2007 resulting in 11.7ha of moorland being rewetted at Blackpitts and a further 12.3ha at other locations within the unit (David Smith, personal communication.). Characteristic features of the *Blackpitts 2* transect are the two pools that have been created, which cross the transect at 13m and 40m. Each one is roughly 5m x 15m with a depth of around 2m when full.

(b) Vegetation survey

There were two previous vegetation surveys along the *Blackpitts 2* transect - one in August 2006 prior to rewetting works, and another in August 2008, after rewetting was completed. To further the data set and assess the current floral biodiversity, another survey was undertaken on 3rd September 2009, along the same transect. Initially, the end of the transect was located by identifying existing marker pegs in the ground and then a tape measure was laid along the ground starting from these pegs and extending 50 metres eastwards. To define the sampling areas, a 1 metre squared quadrat was used. It was divided into 4 equal sections (of 25cm²), named *i*, *ii*, *iii*, and *iv*, which corresponded to the data recording sheets. Starting at 0 metres on the tape measure, the quadrat was laid...
down next to the tape measure (on the south side) every metre and the vegetation within recorded. Within each quarter of the quadrat, if a species was present, regardless of its abundance, then the corresponding boxes were marked as ‘1’ on the recording sheet, so that the total abundance for any given species in a quadrat was 4. In order to find every species present in each quadrat, thorough searching by hand was required especially as some of the grasses are long and hide smaller plants that grow close to the ground. In the case of quadrats being in pools, any vegetation present was recorded as accurately as possible, often by eye from a distance but, due to health and safety risks, detailed examination of vegetation in pools was not carried out. Identification books were used to aid recognition of plants. The raw data was then input into a corresponding excel spreadsheet which allowed calculations and statistical analyses to be performed.

(c) Dipwell installation and monitoring
In order to investigate the relationship between water table and vegetation composition, dipwells were constructed and installed along the Blackpitts transect. In a method similar to that in Allott et al. (2009), polypropylene waste pipe with a diameter of 2cm was cut into lengths of 75cm, 100cm and 150cm. Holes (2mm in diameter) were drilled on either side of each tube at 10cm intervals to allow water infiltration below ground. A line was marked around each tube 10cm from the top to show that this section should be left protruding above ground. Two ‘air holes’ were drilled within this section to minimise any effect on water level from potential imbalances of air pressure inside the dipwells, caused by them being sealed. Each dipwell was then placed into one leg from a pair of tights and this was pulled up to the previously marked line and secured with heavy duty waterproof tape. Tights were used as they prevent debris entering and blocking the tubes. The dipwells were strategically located along the transect in a way that enabled changes in water table, especially in the wetter areas, to be detected hence the clustering around pools (figure 1). Peat cores were taken at each dipwell site to determine the depth and stability of ground. Where it was particularly soft, in more waterlogged areas, longer lengths of tube were used to ensure the dipwells stayed stable in the ground. Also, this provided a means for inserting the dipwells into the ground. Most dipwells were installed so that 10cm of tube was left protruding above ground but those in wetter areas more prone to a variable water level were left with longer lengths above ground so that they did not become flooded if water
level rose. Once they were installed, the hole at the top of each tube was covered, again with heavy duty waterproof tape to prevent rainfall entering.

In order to gain reliable values for water table height, recordings from the dipwells were taken (approximately) weekly throughout the study period. A water level meter was used to measure the depth of the water inside the dipwells. It works on the principle of making a bleeping noise when it encounters water. For each dipwell, after the tape ‘lid’ had been removed, the water meter was switched on and was then lowered into tube until the detection-sound was audible. At this point a depth measurement was taken of where the top of the tube was against the cable. The tape was then secured back onto the top of the tube. Also, a measurement of the height of the tube above the ground was taken using a plastic 30cm ruler. For dipwells that had a longer length of tube above ground, notably those around pools, a longer rigid tape measure was used. The ruler/tape measure was pushed down the side of dipwell until resistance from the ground was met, and a value was recorded. For consistency, measurements were always taken on the north facing side of the dipwells as the opposite side was often slightly lower or higher due to the unevenness of ground. For those dipwells in water, the tape measure was inserted into the water until a firm substrate was encountered. A value for water table height of each dipwell was then calculated by subtracting the water depth from tube height, and displayed graphically (figure 1). These values were used for the analysis of the relationship with plants. Rainfall data for the site was also obtained and used to analyse the relationship between precipitation and water table. It can be seen that the water table has remained relatively constant but does respond to prolonged periods of high or low rainfall (figure 2).

![Figure 1](image.png)

Figure 1. Average water table height over study period per quadrat along the Blackpitts 2 transect. Positive values on the y-axis signify water table was above ground, negative values signify water table was below ground. Error bars are equal to 1 standard error either side of the mean, and this indicates water height variability per dipwell.
Firstly, the vegetation community as a whole was analysed to see how it had changed, if at all, since re-wetting. From the raw data sets for each of the 3 years, overall species richness (number of species present), abundance (total number of individuals present) and evenness (Simpson’s measure of equitability) were calculated as measures of diversity, and rank abundance plots were created as a visual representation of the composition of the community. To be able to concisely analysis the entire community and species relationships, a principal component analysis (PCA) was undertaken using the computer software package SPSS (SPSS Inc. Chicago, Illinois, USA). This generated a group of species that consistently occurred together (see appendix 2ii for full output), and this assemblage was used to examine the community response to differing water table heights along the transect. As the restoration project is aiming to increase moss species and reduce the dominance of graminoids, these target groups were then analysed further in

![Figure 2. Water table response to rainfall. Weekly mean value for water table height across the whole transect is plotted with daily rainfall totals over the same period. This could be used in the future to properly analyse water table response to rainfall events and hence how variable or stable the water table is and the effects this has on vegetation.](image)

(Rainfall data is from The Environment Agency.)

(d) Data analysis
order to determine whether the project is having these intended outcomes. The abundance and richness for each group were determined to see change over the three years. Each quadrat was assigned a value for water table from its nearest dipwell and the abundances and richesses from 2009 were used to correlate with water table height in order to examine if wetter areas had higher abundances of moss and lower abundances of grassy species, and hence establish whether raising the water table is having, and will continue to have, the desired outcome on biodiversity. In addition to this, the individual species within each group were plotted against water height to visualise the more specific effects of raising the water table further to the groups as a whole, while contingency tables using Fisher’s exact test calculated if the presence or absence of groups/individual species were significantly related to water table height (see appendix iii and iv for workings).

3. RESULTS

(a) Overall species changes

(i) Species richness
The richness of the whole community has increased since re-wetting but not significantly (one-way ANOVA, $F_{2, 147} = 0.17, p > 0.05$; table 1). There was an increase of seven species between 2006 and 2008, but no further increase in 2009 (table1; figure 3). Twenty species remained constant, eleven have appeared since 2006, eight of which are highly associated with wet habitats. However, three species of moss that had previously been present were no longer found in 2009.

(ii) Species evenness
There has been a slight decrease in evenness since 2006 but generally the community has remained relatively uneven with a clear dominance by the most abundant species, especially *M. caerulea* (table 1; figure 3).

(iii) Abundance
The total number of individuals in the community decreased from 2006 to 2008 but increased again in 2009, the latter year having the highest overall abundance since the restoration project began, but not a significant change (one-way ANOVA, $F_{2,147} = 0.13, p > 0.05$; table 1). The rank abundance plots for each year (figure 3) show the comparative
abundances of the individual species. *M. caerulea* was by far the most abundance species in all years, followed by *E. vaginatum*. After *M. caerulea*, the differences between abundances of species were a lot smaller. The group of the next five most abundant species was shared by all years and consists of *E. tetralix, E. angustofolium, T. cespitosum, Hypnum cupressiforme* and *Narthecium ossifragum*, but their individual rank abundances differed markedly between years. The rest of the composition of the least abundant species differed markedly between years.

Table 1. Diversity measures for the whole vegetation community plus the target groups of mosses and grasses, over time. Abundance was measured as the total number of individuals present, richness was measured as the number of species present, and evenness was calculated using Simpson’s measure of equitability.

<table>
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<tr>
<th>Diversity Measure</th>
<th>Year</th>
<th>P-value</th>
</tr>
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<tbody>
<tr>
<td></td>
<td>2006</td>
<td>2008</td>
</tr>
<tr>
<td><strong>Abundance</strong></td>
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<td></td>
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<tr>
<td>All species</td>
<td>1167</td>
<td>1135</td>
</tr>
<tr>
<td>Mosses</td>
<td>210</td>
<td>255</td>
</tr>
<tr>
<td>Graminoids</td>
<td>590</td>
<td>526</td>
</tr>
<tr>
<td><strong>Species richness</strong></td>
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<tr>
<td>Mosses</td>
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<td>12</td>
</tr>
<tr>
<td>Graminoids</td>
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<td>9</td>
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<tr>
<td><strong>Species evenness</strong></td>
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<tr>
<td>Graminoids</td>
<td>0.58</td>
<td>0.50</td>
</tr>
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</table>
Figure 3. Rank abundance plots showing community composition in (a) 2006. (b) 2008. (c) 2009.
(iv) **Species assemblages**

In order to examine the interactions between species instead of the community as a whole, a principal component analysis was conducted on the data. Principal component 1 (PC1) accounted for 12.3% of the total variance of the data set. The rest of the principal components accounted for around 3% or less and therefore it was deemed necessary to interpret PC1 only. Looking at the rotated component matrix, the species most positively correlated with PC1, and therefore were the assemblage that occurred together, were the following: *T. cespitosum, E. tetralix, N. ossifragum, E. angustifolium, E. vaginatum* and *C. vulgaris*. Most of these species happen to also be in the group of most high abundance. The negative correlates were not considered here as they were all only very weakly correlated. The output gave scores for PC1 for each quadrat showing the degree to which each one contained the species related to PC1. The average PC1 score over all the quadrats has decreased since 2006 but not significantly (one-way ANOVA, $F_{2, 147} = 1.11, \ p > 0.05$; figure 4a) (See appendix 2ii for full PCA output).

(v) **Relationship with water**

When the PC1 scores from 2009 were correlated with water table height, there was a significant negative correlation (Pearson’s correlation coefficient, $r = -0.49, \ df = 48, \ p < 0.01$; figure 4b), with quadrats that had higher scores (meaning they contained more of the species positively related to PC1) being found where the water table was low, and quadrats that contained these species to a lesser degree were in areas of higher water table. It is clear looking at the raw data that since rewetting abundances per quadrat have generally increased, while species richness per quadrat is more variable.

(b) **Mosses**

(i) **Richness and abundance**

Similarly to the richness of the whole community, the richness of mosses increased between 2006 and 2008 but has remained the same, twelve, since then. This change is not significant (one-way ANOVA, $F_{2, 147} = 2.67, \ p > 0.05$; table 1). Five species have remained constant, seven species have established since 2006, whereas three species (*Aulacomnium palustre, Dicranum scoparium*, and *Sphagnum denticulatum*) appear to have been lost since 2008. Abundance has increased year on year, but not significantly (one-way ANOVA, $F_{2, 147} = 2.36, \ p > 0.05$; table 1). *H. cupressiforme* has always been the
dominant moss species, but whereas it increased slightly in abundance between 2006 and 2008, its abundance has now dropped to just below 2006 levels. Another highly abundant species, *S. cuspidatum*, has increased substantially to 63 occurrences in 2009 compared to nine originally.

(ii) Relationship with water

Neither richness nor abundance of mosses in 2009 was related to water table height (figures 5a & b). It was determined that both the presence and absence of mosses were relatively equal in that they were present at all water depths and absent at all water depths, and the relationship between their abundance and water table height was therefore not significant (Contingency table using Fisher’s exact test, \( p > 0.05 \); table 2; figure 5b). The relationship between water and individual species of mosses varied considerably. The presence of *H. cupressiforme* was significantly linked to lower water height, and was absent mostly waterlogged areas (Contingency table using Fisher’s exact test, \( p < 0.01 \); table 2; figure 5e). A similar but stronger relationship was shown by *S. tenellum* (Contingency table using Fisher’s exact test, \( p < 0.001 \); table 2; appendix 2iii) but in contrast, *S. cuspidatum* presence was significantly related to a higher water table (Contingency table using Fisher’s exact test, \( p < 0.05 \); table 2; figure 5f), being consistently observed in the bog pools. *S. palustre* showed the same pattern but was not significant (Contingency table using Fisher’s exact test, \( p > 0.05 \); table 2; appendix 2iii). Looking at the abundances of the other species, many showed highest or most frequent abundances at a water table depth of between 0 and -20 cm and where absent, were so along the whole range of depths (see appendix 2iii).

(c) Graminoids

(i) Richness and abundance

The richness of these groups has remained relatively constant, initially increasing from seven in 2006 to nine in 2008, but the 2009 survey found there to be eight. The change between the years is not significant (one-way ANOVA, \( F_{2, 147} = 2.12, p > 0.05 \); table 1). Although there has been an overall decrease in abundance since 2006, there was a decrease between 2006 and 2008 followed by an increase to the present level. This change was again not significant (one-way ANOVA, \( F_{2, 147} = 0.73, p > 0.05 \); table 1).
Looking at the abundances of some key individual species, the amount of *M. caerulea* along the transect has decreased from 200 occurrences in 2006, to 175 in the following years. Despite an increase of *E. vaginatum* in 2008, its overall abundance has fallen from 148 occurrences in 2006 to 137 in 2009.

(ii) Relationship with water

Species richness of these groups had no relationship with water table height (figure 5c). The abundance of these groups did however have a strong negative relationship with water table height, higher levels of water being associated with less graminoids being present (Contingency table analysis using Fisher’s exact test, $p < 0.05$; table 2; figure 5d). The key species *M. caerulea* and *E. vaginatum* were both highly abundant across all water levels, but whereas *E. vaginatum* was also absent at the entire range of water table levels (Contingency table analysis using Fisher’s exact test, $p > 0.05$; table 2; figure 5h), *M. caerulea* was only absent in locations of above-ground water table, and this finding was significant (Contingency table analysis using Fisher’s exact test, $p < 0.05$; table 2; figure 5g). Of the other species sufficiently abundant to be tested, *E. angustifolium* was significantly linked to being present at higher water levels but zero-abundances were found everywhere (Contingency table analysis using Fisher’s exact test, $p < 0.05$; table 2; appendix 2iii), and *T. cespitosum* was mostly present in conditions of low water and absent at high water levels (Contingency table analysis using Fisher’s exact test, $p < 0.05$; table 2; appendix 2iii). Of the other species in this group, most showed a pattern of being present most often between depths of -5 to -15cm, and where absent, were so at the whole range of depths (see appendix 2iii).

Figure 4. PCA results. (a) Average PC1 score for the 3 years of study. Error bars show 1 standard error either side of the mean. (b) The relationship between PC1 score and water table height. Error bars are equal to 1 standard error either side of the mean, $r = -0.49$. 

![Figure 4](image-url)
abundance against water table height. (a) Mosses species richness. (c) Graminoids species richness. (d) Graminoids abundance. (f) S. cuspidatum abundance. (g) M. caerulea abundance. All data is from 2009 only.
Table 2. Link between higher/lower water table height. H signifies a species/group is present or absent at higher water table heights (≥ -8.7cm), L signifies a species/group is present or absent at lower water table heights (< -8.7 cm), E signifies that a species/group is present or absent relatively equally at higher and lower water table heights. The boundary between higher and lower was determined by taking the median value for water table height (-8.7cm). ns denotes no significance, * denotes $p < 0.05$, ** denotes $p < 0.01$, and *** denotes $p < 0.001$. Species were only tested if they had an abundance of 10 or more, in order to see patterns. See appendix 2iii for associated contingency tables.

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<th>Absence</th>
<th>Significance</th>
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<td>ns</td>
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<tr>
<td><em>H. cupressiforme</em></td>
<td>L</td>
<td>H</td>
<td>**</td>
</tr>
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<td><em>S. capillifolium</em></td>
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</tr>
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<td><em>S. cuspidatum</em></td>
<td>H</td>
<td>E</td>
<td>*</td>
</tr>
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<td>*</td>
</tr>
<tr>
<td><em>T. cespitosum</em></td>
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<td>H</td>
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4. DISCUSSION

As expected, the overall vegetation community along the Blackpitts 2 transect has changed in response to rewetting, and is showing signs of progression to a healthier, more diverse ecosystem, which is the aim of the restoration. While the species composition has
remained uneven, richness and overall abundance of vegetation has increased. Encouragingly, the abundance and richness of the mosses have increased, but whereas some new species have appeared, a three have disappeared. Another target group, the graminoids, which the project is hoping to reduce the dominance of, have decreased in abundance but the number of species in this grouping has remained relatively constant. In particular, *M. caerulea*, which previously dominated the site, still prevails as the dominant species although its abundance has dropped slightly since rewetting, which is again a promising sign of change. With regard to water table, in general, both target groups were found at the full range of water table depths. While there was no relationship between moss abundance and water table height, the graminoids and the PC1 assemblage (which consisted predominantly of grasses, sedges and rushes) were negatively associated with a higher water table, indicating that in particularly waterlogged conditions, the grass-like species are unable to thrive, unlike the mosses which were abundant in these conditions.

The widespread distribution of the graminoids can be explained by that fact that, while graminoids are often seen to be dryland plants, the ones observed here are adapted to living in soils with a high water content, for example transporting gas from the shoots to underground waterlogged tissues and roots in order to ventilate them (Armstrong *et al.* 1991, in Brix & Sorrell, 1996), as well as accumulating carbohydrates to allow tissues to respire anaerobically (Brix & Sorrell, 1996). *M. caerulea* is particularly well adapted to a varying water table. It can grow in drier areas like other typical grasses but is also adapted to waterlogged conditions by having aerenchyma (air channels) in its roots which contains 15-20% oxygen gas (Webster, 1962). Also, creation of an oxygenated zone around the rhizosphere acts effectively as a buffer zone between roots and the anaerobic environment (Taylor *et al.*, 2001). However, the general trend for each group does not hold true for all species; there are many different species-specific relationships, each distinctively adapted to the moisture content of its niche. For example, *T. cespitosum, A. odoratum* and *Agrostis* species were more abundant at lower water levels (generally < -4.2cm) and not at higher levels, which is characteristic of these species (Grime & Lloyd, 1973), but *Eriophorum* species and *M. caerulea* were present throughout the entire range of water depths although their abundances were low in the bog pools. While it may necessary to reduce the dominance of the graminoids in order to create a more diverse community, they are still important for the correct functioning of the ecosystem. For example, commensalism can
occur between *Sphagnum* mosses and *E. vaginatum* (Grosvernier et al., 1997) and *E. angustifolium* (Ferland & Rochefort, 1997). These sedges have been documented as improving unfavourable local conditions for the mosses, protecting their diaspores (seed dispersal units). *M. caerulea* is also an important species. Despite being viewed as an invasive pest on Exmoor, ‘purple moor grass and rush pasture’ is a targeted habitat under the UK Biodiversity Action Plan (UKBAP) as much of it is threatened by land use changes (UK Biodiversity Steering Group, 1995), and so while it may be desirable to enhance the blanket bog habitat on Exmoor, which is also a UKBAP target, the correct balance is needed to ensure that not too much *M. caerulea* displaced.

Within the mosses, many of the *Sphagnum* species, in particular *S. cuspidatum, S. papillosum, S. fallax*, and *S. palustre*, were more abundant at higher water than lower water table, conforming with previous findings that the *Sphagnum* species and in particular *S. cuspidatum* are restricted to water-filled hollows (Nordbakken, 1996). A previous study showed that *S. cuspidatum* and *S. fallax* have higher productivity and growth in these conditions as this is their optimal niche (Grosvernier et al., 1997), but other *Sphagnum*’s, notably *S. subnitens* and *S. tenellum*, occurred more often at lower water table heights, as did *H. cupressiforme* and *Campylopus paradoxus*. This was expected by the latter two species as they commonly inhabit areas of lower moisture like trees and rocks (Dixon & Jameson, 1904). The emergence of seven new moss species since 2006, plus *Drosera rotundifolia* which is widely known as a wetland plant (Elkington et al., 2001) implies that wetland biodiversity has increased. Conversely, three moss species have disappeared in the same time period. One of the species that appears to be no longer present is *D. scoparium*. As this moss is generally found in drier habitats, for example woods and heaths (Dixon & Jameson, 1904) the loss suggests that water table has increased and has remained high. But the apparent loss of water-loving *S. denticulatum* and *A. palustre* on the other hand was unexpected. However, as these species were previously only found in low abundances and in only a few quadrats, they may well still be there but were missed due to human error during the present survey.

There are many other errors in experimental design and implementation that have provided limitations to this investigation. Firstly, a major limitation with the vegetation survey was there was no estimation of cover of plants, only if they were present or not. A more accurate method would be to estimate the percentage cover of each species within a
quadrat by eye, or using a point-frame quadrat which is a useful tool for this (Evans & Love, 1957). Another questionable factor is the size of the sample used. Quadrats can be ineffective at measuring precise numbers of plants, but used at high intensity can detect accurate species richness (Jorgensen & Tunnell, 2001). The vegetation results here can be deemed as accurate because the whole of the transect was surveyed, not just parts of it. However, to be able to evaluate vegetation change in the area post-rewetting, it would be better to survey a wider area, perhaps 100m$^2$ or more, as this should encapsulate more species. In this case, a pilot study should be carried out to determine the number of quadrats to be used in order to achieve 90% or more detection of all species (Sparks et al., 1997). However, for the purpose of studying water table in regard to vegetation composition, the method used here was effective as the vegetation samples were taken directly along the dipwell line. It would have been better to install dipwells every metre so that each quadrat had a direct water table value instead of some quadrats relying on a value from a dipwell a few metres away.

There were no measurements of water table height at this site previous to rewetting, so it is not possible to quantify the raise in water table or say how low it was previously. However, Price & Whitehead (2001) suggested that a high water table can be considered as -24.9cm +/- 14.3 cm. This means that the water table at the Blackpits 2 transect (average -7.8cm with a range of -22.6 – 9.9cm) can be considered high and the implication of this is that when talking here about a ‘lower’ water table (< -8.7cm, which was the median value), this is actually a high water table.

Although this study provides an insight into water table height in relation to vegetation composition, there are many other environmental variables acting as confounding factors that also influence vegetation establishment and growth. Of particular importance is water table fluctuation, for example Rutter (1955) found that there is more *M. caerulea* in areas of higher water table fluctuation, possibly due to the flushing of toxic substances and renewal of nutrients. It has also been found that *M. caerulea* has a higher growth rate with increased levels of phosphorus (Taylor et al., 2001), as do other vascular plants and some *Sphagnum* species (Rochefort et al., 2003), showing that the chemical composition of peat can influence plant growth. These factors should therefore be considered and are worth investigating in the future.
Thinking about restoration overall, Buckland et al., 2000 (in Goreham & Rochefort, 2003) suggest that it might not be possible to restore a mire to its original condition as many original Sphagnum species may have become extinct. Also, determining the success of a restoration is confounded by the fact that it takes a long time for an ecosystem to progress to a new state. In the case of mires, characteristic species appear in three to five years, but a functional ecosystem takes around thirty years to establish (Goreham & Rochefort, 2003). Nevertheless, two years after rewetting, blanket bog-associated species have established and it is clear that in the areas of highest water table, NVC type M2/M3 bog pool communities have been created and thus succession to a healthy blanket bog community is progressing.

This study has provided useful information about the current level of vegetation change post-rewetting and can be used as a comparison in future years to examine further changes. It will be necessary to monitor vegetation change and water table variables over a long term period to attain a definitive outcome of restoration, but at this stage it appears successful. The dipwells can be continually monitored in order to gain a larger data set which will provide more reliable estimates of water table height and variability, and these can be used along with analysis of other factors that influence vegetation distribution in order to gain a wider knowledge of environment-plant dynamics.

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